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Are natural floods accelerators for streambank vegetation development in floodplain restoration?

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Abstract

Riverbanks are very dynamic habitats for riparian vegetation strongly influenced by fluvial and geomorphic processes. This habitat type was severely reduced in the past by river straightening and bank stabilisation. Restoration and establishment of new floodplain streams promote this habitat, but a directed succession to later stages was observed many times. Our study aimed to analyse whether the often observed directed succession of the streambank vegetation after restoration implementation could be reversed by a natural flood along a newly created floodplain stream. We investigated the effects of a natural flood in 2013 and different prerestoration conditions on species development in the riparian zone. Vegetation was studied along 12 transects in four different sections from 2011 to 2014. Species composition differed strongly between the sections. Species richness was lowest in a newly dug steep section with high morphological dynamics and highest on wider flat streambanks. Changes during the years reflecting different hydrological events varied between sections. The high natural flood in 2013 reduced the cover of the herb layer and increased bare ground, which led in most sections to a loss of non-target species. Total target species richness did not change due to the natural flood, while target species showed a high turnover rate. In the following year, however, the flood-induced development of species composition, in general, was reversed. Natural floods changed abiotic and biotic conditions along the streambank, but they did not accelerate ecological restoration towards predefined target ecosystems. However, they were necessary to preserve the needed dynamic vegetation changes and species turnover to hinder the succession to later stages dominated by a few species. Our study shows that riparian vegetation near the streambank can be monitored most effectively in cross-profile transects, both in the long-term and event-related.

KEYWORDS

controlled discharge, monitoring, riparian vegetation, species composition, species turnover

Abbreviations: NMDS, non-metric multidimensional scaling; PERMANOVA, permutational multivariate analysis of variance.

1 | INTRODUCTION

Rivers and their floodplains in Europe and North America are strongly anthropogenically influenced and their flooding regimes and morphological characteristics have been drastically changed during the last centuries (Globevnik et al., 2020; Nilsson et al., 2005). Many rivers have been regulated from braided or anastomosing rivers to single-thread rivers with relatively stable flow conditions apart from irregular high floods (Tockner et al., 2010). They were forced into a riverbed and their banks were straightened and fixed in a steep and fortified manner (e.g., with ripraps; Wollny et al., 2019). While floodplain habitats, in general, suffered from missing regular floods, the vegetation directly adjacent to the watercourse, in particular, was significantly reduced in its expansion (one main channel in contrast to several side-arms). Furthermore, riverbank habitats lost their typical dynamic characteristics of regular hydrological and morphological disturbances (floods, low water, erosion and accumulation). Thus, the number of typical plant species in this habitat type (Harvolk et al., 2014; Nilsson & Svedmark, 2002; Wollny et al., 2019) and its ecological functionality concerning nutrient cycling, temperature regulation, bank stabilisation and as habitat for specialised target species (González et al., 2015) were significantly reduced.

A common and effective restoration approach for smaller rivers and their floodplains is the widening of the river channel (González et al., 2015; Göthe et al., 2016; Jähnig et al., 2009; Modrak et al., 2017; Rohde et al., 2005), which effectively increases the width of the banks and the area of riparian habitats. However, along larger rivers, the entire river and its floodplain can often not fully be restored due to numerous strong restrictions (e.g., hydropower, navigation or settlements). In this case, it can be promising to reconnect smaller floodplain streams or secondary channels with the main stream (Meyer et al., 2013; Schiemer et al., 1999; Staentzel et al., 2018). Mostly, only a small part of the main river's water can be diverted to still meet the requirements of society regarding the river itself (e.g., energy production, provision of waterway for navigation), while returning to predisturbed conditions is rarely possible (González et al., 2015). Nevertheless, floodplain restoration aims to re-establish floodplain-typical conditions and habitats for target species communities (i.e., fluctuating water zones, softwood forests, reed beds and tall herb vegetation). In this context, enhanced fluvial dynamics are necessary to initiate the re-establishment of floodplain target plants (Dufour & Piégay, 2010; Nilsson & Svedmark, 2002).

Positive short-term effects of constructive restoration measures on riparian vegetation often decline without ongoing fluvial or morphological disturbance (Bauer et al., 2018; Januschke et al., 2014). While extraordinary floods have a great impact on the occurrence of geomorphic features and their target species (Hering et al., 2004; Ilg et al., 2009; Negishi et al., 2019), intermediate floods (e.g., decennial events) still determine plant community patches and the occurrence of certain target species (Džubáková et al., 2015; Nilsson & Svedmark, 2002; Renöfält et al., 2005). Januschke et al. (2014) and Lu et al. (2019) have highlighted the need for regular long-term monitoring, especially in dynamic floodplains. However, it might be promising to additionally study the effects of typical but higher flood events in detail, as these may be more important for community composition of the riparian vegetation than long-term successional processes. In addition, it is challenging to monitor the streambank vegetation in dynamic floodplains, as this linear habitat is spatially ever-changing due to erosion and accumulation (Lang et al., 2013).

This study aimed to explore vegetation development in the direct vicinity of a reconnected and partly newly built floodplain stream of 8 km length in the Danube floodplain (Stammel et al., 2012). The stream was created in a still species-rich but rarely flooded floodplain with different distinct hydrological sections before restoration (no water, temporary lentic or lotic, lotic conditions; Schwab et al., 2018). Three years after the opening of the floodplain stream, a natural flood occurred. Thus, we did not investigate the ongoing development after restoration here, but the impact of a decennial flood event on target species groups of the linear streambank vegetation. Inherently, permanent plots on the banks can be destroyed and disappear by geomorphic processes over the years. Cross profiles, in contrast to stratified permanent plots, are an important method to observe the unpredictable development of morphodynamically active riverbeds (Roni, 2005) and can also be used for vegetation surveys. Our research aimed to analyse whether the often observed directed succession of the riparian vegetation to later stages after restoration can be reversed by a natural flood. We addressed the following questions:

- What is the effect of a decennial flood on plant target groups in a newly created floodplain stream and does this effect persist in the following year?
- Is this effect homogenous between sections, or are there differences depending on prerestoration conditions?
- Which conclusions for efficient vegetation monitoring along morphodynamic rivers can be drawn from the results?

2 | METHODS

2.1 | Study area

The study area is in the floodplain of the River Danube in Germany between Neuburg/Donau and Ingolstadt (river km 2473–2464, 48°45′N, 11°16′E; Figure 1). An oceanic climate dominates with a mean annual temperature of 8.8°C and annual precipitation of 744 mm (1981–2010; Fischer, 2016). Soils consist of overlaying calcareous, nutrient-rich substrates of mainly fine-grained alluvial loam (Fischer, 2016). The Danube was subject to significant anthropogenic changes over the last two centuries to ensure flood protection, land reclamation and hydropower use. The river was straightened and embanked, the water level equalised by dams and the connection between floodplain and river interrupted (Fischer & Cyffka, 2014).

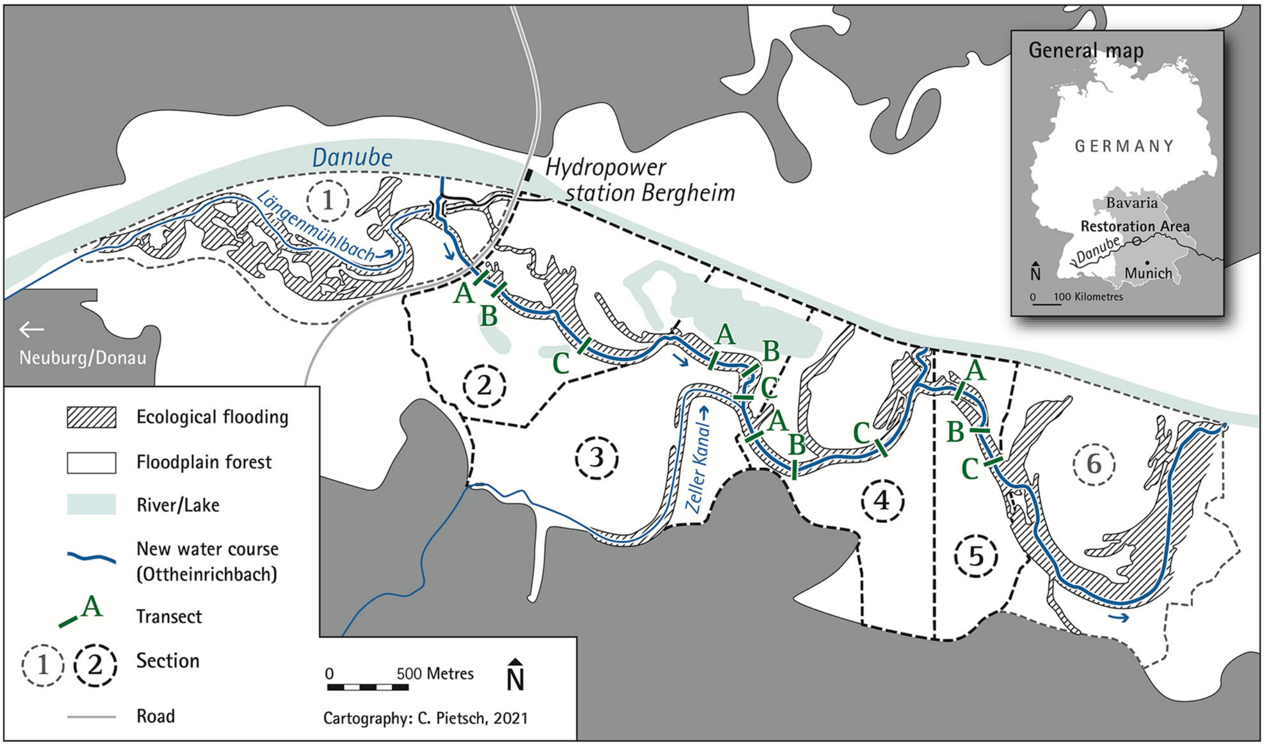

FIGURE 1 Location of the study area (small map) and division of the new floodplain stream into six sections, of which Sections 2–5 were investigated. Sections represent different hydrological conditions before and during restoration and different inclinations of the streambank (see Table 1)

Hence, the typical anastomosing self-development of the Danube riverbed was prevented. In the floodplain, a few backwaters remained with a constant water level and neglectable connectivity to the Danube. A small stream, the Zeller Kanal (Figure 1), runs through parts of the floodplain into the Danube.

2.2 | Floodplain restoration and flood duration

A large-scale floodplain restoration project was then realised within the continuous alluvial, rarely flooded forest (for details, see Stammel et al., 2012). The project aimed to restore the floodplain by enhancing fluvial dynamics and improving lateral connectivity. Therefore, a new permanent watercourse of about 8 km length was created. The controlled dynamic discharge of the new stream (1–5 $m^3 s^{-1}$ and up to 30 $m^3 s^{-1}$ during ecological floodings) is adapted to the discharge of the Danube (Stammel et al., 2012). Within the first 2.5 km, a new channel was dug into the alluvial sediments along existing dry channels. In contrast, the following part of the stream runs through already existing (temporary) water bodies and subsequently merges with the temporary stream Zeller Kanal.

The new floodplain stream was opened in 2010. A first ecological test flooding with a lower total discharge of 15 $m^3 s^{-1}$ for 1.5 days was conducted in 2011, followed by two ordinary ecological floodings in 2012 (max. 30 $m^3 s^{-1}$ for 5 days). In 2013, seven ecological floodings (total duration 24 days) occurred and one natural flood with a maximum discharge of c. 200 $m^3 s^{-1}$ and a duration of 8 days in June 2013 (Fischer, 2016). In 2014, no ecological flooding was diverted.

2.3 | Study design

The data described in this paper are part of a long-term monitoring programme to enable adapted management, which started before the restoration in 2008 (Stammel et al., 2012). Here, we present the development of the streambank vegetation for the years 2011–2014. For monitoring purposes, the watercourse was divided into six sections (Figure 1), covering the hydrological conditions before the restoration and the planned restoration measures (Lang et al., 2013; Schwab et al., 2018). For our investigation, we excluded Sections 1 and 6 due to their considerably different fluvial characteristics (lotic conditions, fed by another stream). Thus, four sections (2–5) were selected (Table 1). The mean low water level was measured each year by diverting exactly 1 $m^3 s^{-1}$.

In each of the four sections, three transects were installed orthogonally to the stream covering the entire width of the regularly flooded area. Transects (width: 1 m) of 21–57 m length were subdivided into plots of 1m × 1 m. For this study, we only investigated the plots situated between the measured mean low water level and 1 m above this level (158 plots from 12 transects) and defined this part as the 'riparian zone'. Due to morphological changes and the

TABLE 1 Abiotic conditions in the selected study sections and expansion of the 12 sampled transects

	Section			
	2	3	4	5
Discharge of new stream ($m^3 s^{-1}$)	1–5	1–5	1–5	0–2.5
Hydrological condition before restoration	No water	Temporary lentic	Lentic	Temporary lotic
Impact of construction work	High/new built	Medium	None	None
Inclination	Steep	Flat	Medium	Medium
Length of entire transects (m)	2A: 27	3A: 57	4A: 34	5A: 21
	2B: 23	3B: 45	4B: 43	5B: 31
	2C: 31	3C: 57	4C: 49	5C: 30
Number of plots in the riparian zone (min–max of all years)	2A: 5–6	3A: 13–20	4A: 5–7	5A: 11–13
	2B: 7–12	3B: 14–16	4B: 12–14	5B: 14–15
	2C: 5–10	3C: 21–24	4C: 6–9	5C: 11–12

changing water level, the number of plots in the riparian zone slightly varied per transect and year.

2.4 | Field methods

Vegetation relevés were carried out for each 1 m^2 plot of the transects. The percentage cover of all vascular plant species was recorded in August and September of each year using the decimal scale of Londo (1976). Plant nomenclature was based on Buttler et al. (2018). The percentage cover of tree, herb and litter (including deadwood) layers, as well as the percentage of bare ground and water was estimated. The mean plant height of the herb layer was measured. The inclination of each plot was calculated as the difference between the highest and lowest level along the transect. The level was obtained as distance to a stretched line on calibrated permanent poles.

2.5 | Data analysis

For statistical analyses of the effects of sections and years on structural parameters and species richness, we randomly selected five 1 m^2 plots of the riparian zone per transect (*n*: 240; 5 plots × 3 transects × 4 sections × 4 years) and used a nonparametric Kruskal–Wallis–ANOVA (analysis of variance). Differences between the subsets were identified by using the Dunn-test with Bonferroni correction. Statistical analyses were performed using IBM SPSS Statistics 24.

To assess restoration success, target species typical for plant communities of floodplain habitats were defined (Schwab & Kiehl, 2017). Therefore, all species were assigned either to one of the five floodplain habitats (permanent water, fluctuating water zone, reed bed and tall herb stands, softwood riparian forest and hardwood riparian forest) or no floodplain habitat (nontarget species). The number of species (target species as indicators of the above-mentioned habitats, nontarget species, total number of species) of all plots of the riparian zone of each transect was compared between years by identifying newly occurring and missing species (cf. Januschke et al., 2014) and calculating their balance. We analysed species composition by nonmetric multidimensional scaling (NMDS) using the square-root transformed relative frequency of species (distance measure: Bray-Curtis) (PCORD 6.13; McCune & Mefford, 2011). To show the temporal development resulting from the 2013 flooding, we drew the trajectories between the years 2012–2013 and 2013–2014 centred to the origin. A two-way permutational multivariate analysis of variance (PERMANOVA) based on Bray-Curtis dissimilarities was applied to test whether there were significant differences in species composition between years and sections.

3 | RESULTS

3.1 | Morphological and structural differences

Morphological changes between 2011 and 2014 varied significantly between transects of different sections. In Section 2, bank erosion of more than 1 m in depth and 5 m in width occurred, induced by the natural flood and the opening of the new watercourse (Figure 2). Considering all riparian plots of a transect, four new riparian plots were found in transect 2B, while three riparian plots vanished in transect 2C due to erosion. In Sections 3–5, small-scale morphodynamic processes were only observed in the streambed (examples: transect 5A and 5C in Figure 2), but not in the riparian zone.

The inclination of the streambank was significantly higher in Section 2 than in the other sections and significantly lower in Section 3 compared to Sections 4 and 5 (Table 2). Vegetation structure also differed significantly between Section 2 and the other sections. In Section 2, the cover of bare ground was highest (mean for all years: 80%) and the cover and height of the herb layer, as well as

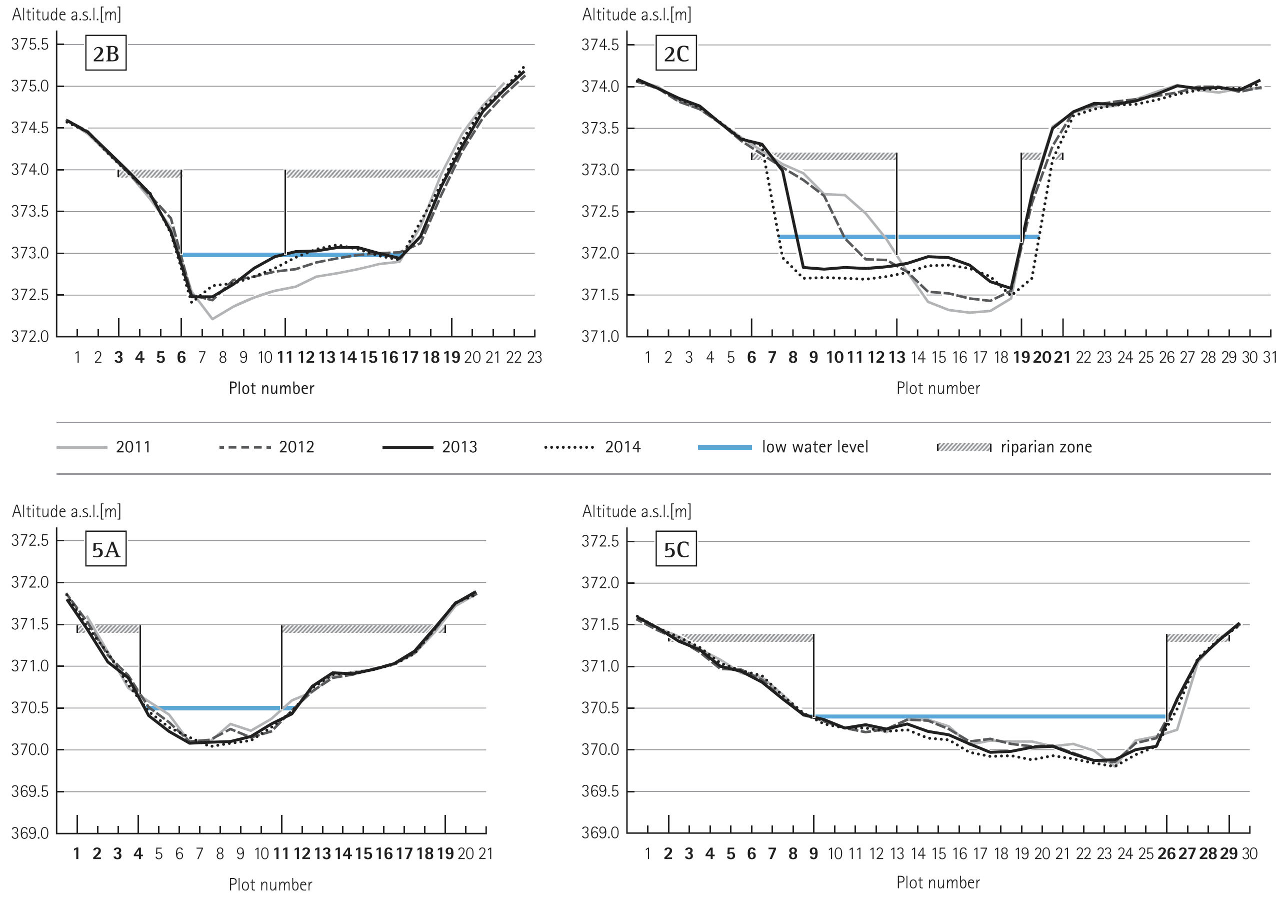

FIGURE 2 Cross profiles of the four transects 2B, 2C, 5A and 5C over the years 2011–2014. Analysed plots of the riparian zone (hatched bar, plot number in bold print) were situated between the low water level (blue) and 1 m above. The *y*-axis is fourfold superelevated to visualise the morphodynamic changes

TABLE 2 Mean values of structural parameters of the different sections across years

		Section			
	Significance	2	3	4	5
Inclination (cm m^{-1})	<0.01	0.43^a	0.15^c	0.31^b	0.20^b
Vegetation height (cm)	<0.01	25^a	60^c	50^{bc}	45^b
Cover herb layer (%)	<0.01	20^a	48^b	33^b	45^b
Cover litter (%)	<0.01	5^a	20^b	30^c	15^b
Cover bare ground (%)	<0.01	80^a	45^b	45^b	40^b

Note: Nonsignificant differences ($p > 0.05$) between single sections are indicated by equal letters behind the values within one row.

litter cover, were lowest (Table 2). Additionally, vegetation height was significantly higher in Section 3 than in 5 and litter cover was higher in Section 4 than in 3 and 5.

Between years, the cover of bare ground differed significantly (Figure 3). In 2013 (natural flood), it reached 70% and was significantly higher than in the years before and after (2011: 48%, 2012 and 2014: 40%). Herb-layer cover was highest in 2012 (mean: 46%) and lowest in 2013 (mean: 34%) ($p = 0.017$, U test). Flood effects, however, differed between sections. In Sections 2 and 4, herb-layer cover decreased significantly due to the natural flooding in 2013, but it remained relatively constant in Sections 3 and 5 (Figure 3). In Section 5, in contrast, herb-layer cover had already decreased from 2011 to 2012.

3.2 | Effects of prerestoration conditions and natural flooding on target and nontarget species

The mean number of 7.5 species per 1 m^2 plot did not vary significantly between years or sections. The mean number of target species per plot, however, was significantly lower in Section 2 (1.6 species) compared to Section 4 and 5 (2.6 and 2.9 species) and Section 3 (4.3 species). The low number in Section 2 was related to a very low number of reed-bed species and indicators of fluctuating water zone. Between years, the number of target species was significantly higher in 2012 than in all other years.

Total species richness, the number of target species and numbers of new and lost target species per transect differed significantly between sections (Section 2: lowest values, Section 3: highest values, Table 3). In Section 2, species typical for hardwood forests dominated the target

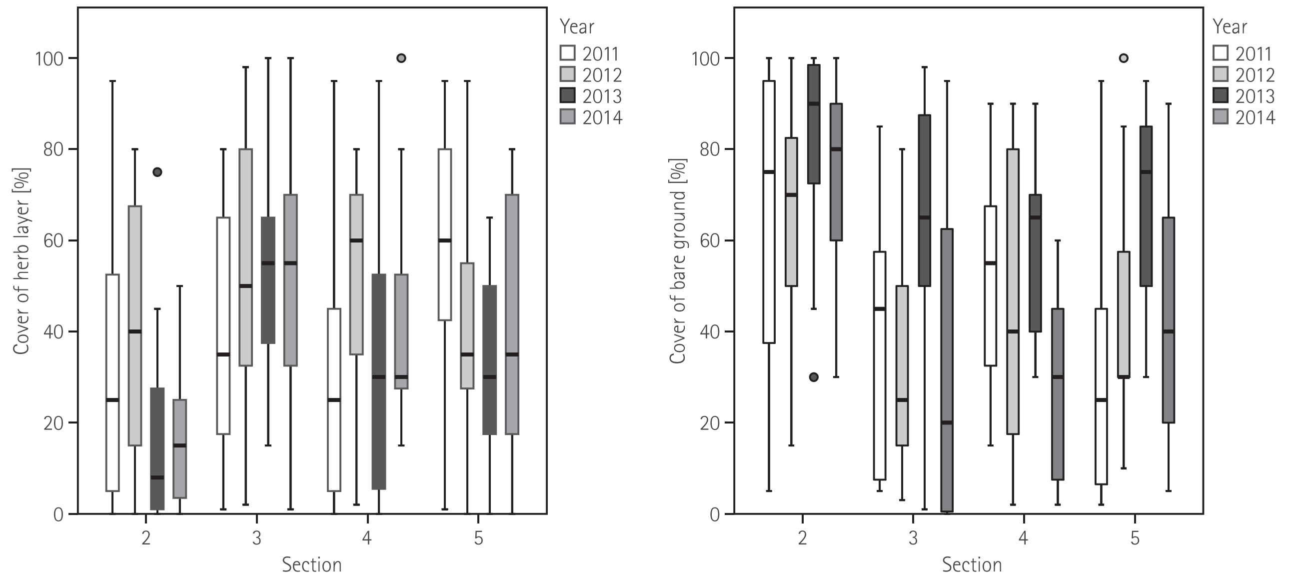

FIGURE 3 Cover of the herb layer (left) and bare ground (right) for the single years for 1 m^2 plots in each section. Boxplots show the median, quartiles, minimum and maximum. Circles indicate the outliers

TABLE 3 Mean of target species and total species number per transect for the different section

		Section			
	Significance	2	3	4	5
Total number of species	<0.01	22.6[a]	48.0[c]	31.3[b]	32.2[b]
Total target species	<0.01	6.0[a]	24.4[c]	13.3[b]	14.8[b]
Species of… (% of all target species)					
… water habitats	<0.01	22.2[b]	13.0[b]	1.2[a]	5.4[ab]
… fluctuating water zone	<0.01	2.0[a]	27.2[c]	8.5[b]	21.7[c]
… reed-bed habitats	<0.01	5.3[a]	32.6[c]	47.1[b]	31.2[c]
… hardwood forests	<0.01	70.6[c]	27.3[a]	43.3[b]	41.8[b]
New target species	<0.01	3.2[a]	8.4[b]	3.6[b]	4.9[b]
Lost target species	0.04	−2.6[a]	−6.9[b]	−3.0[ab]	−5.3[ab]
Remaining target species	<0.01	3.7[a]	17.4[c]	10.4[b]	10.0[b]

Note: Nonsignificant differences ($p > 0.05$) between single sections are indicated by equal letters behind the values within one row.

species (70.6%), followed by water plants (22.2%), whereas indicators of fluctuating waters or reed beds were nearly absent. In Section 3, in contrast, target species of all floodplain habitat types occurred almost in balance. In Section 5, fewer indicators of water habitats but more indicators of hardwood forests were present. In Section 4, in contrast, the number of reed-bed species was significantly higher than in the other sections and they dominated together with hardwood forest species, while indicators of water habitats and fluctuating water zone were underrepresented. In the whole study, almost no target species of softwood riparian forests were found.

Hydrological events (years) significantly affected the balance (new species—lost species per year) of both target and nontarget species. The balance for target species was positive for 2012 (3.75 species per transect), almost neutral (0.25) for 2013, the year of the natural flood and even negative for 2014 (−2.25). This was related to the significantly lower number of lost target species in 2012 (−2.0) compared to 2013 (−5.4) and 2014 (−5.9). The number of remaining target species was constant between the years (10) and total species richness (33.5 species per transect) did not differ between years. The balance of nontarget species was positive in 2012 (2.3), negative in 2013 (−1.9) and neutral in 2014 (0.2), in accordance with the highest number of new nontarget species in 2012 (5.8 compared to 3.0 and 3.6 for 2013 and 2014). The percentage of indicators of single habitats on the total number of target species per transect did not differ significantly between years.

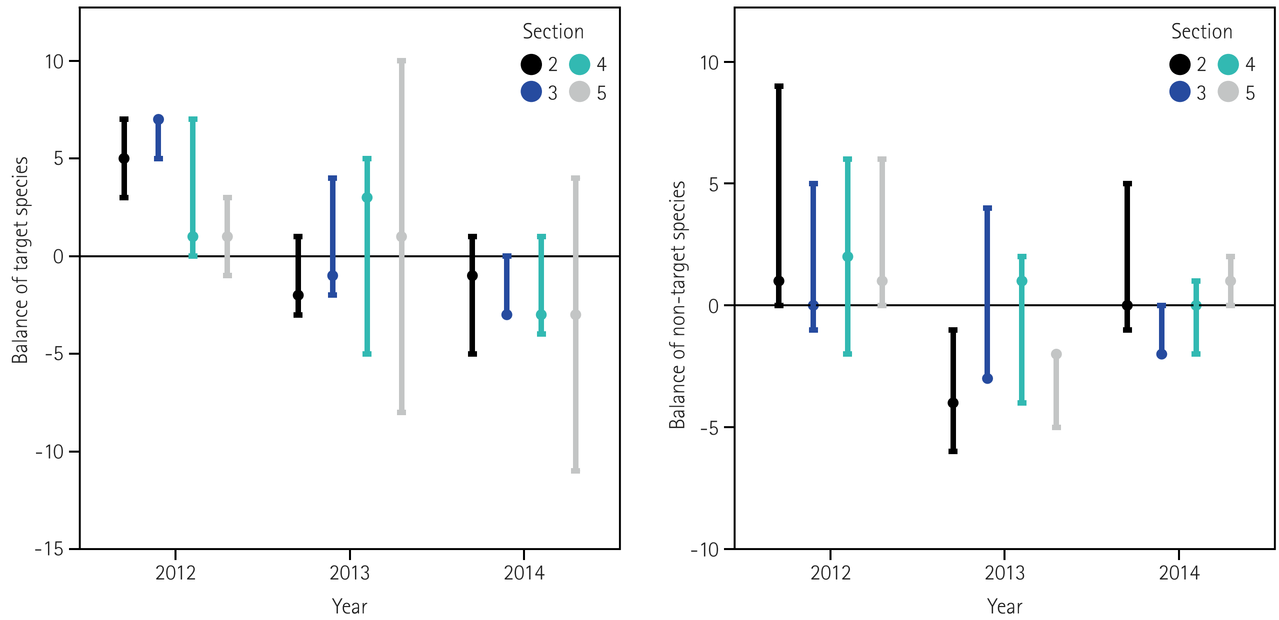

FIGURE 4 Balance of new and lost species (compared to the previous year) of the four sections for 2012, 2013 and 2014. All three transects per section are displayed in the graph either as the maximum, minimum or median. Left: target species, right: nontarget species

In Sections 2 and 3, the balance of target species for 2013 and 2014 was negative (Figure 4). In Section 3, there was an additionally high turnover of species in the different habitat groups. In Sections 4 and 5, in contrast, the number of target species in 2013 increased at four and decreased at two transects. No clear effect of the natural flood on the target species was found, but as a trend, hardwood species and species of water bodies declined in 2013. It was evident that the number of nontarget species declined due to the natural flood in 2013 in nine out of 12 transects and increased again only in four transects in 2014 (no floods) (Figure 4). Not only the number of species but also the cover of reed-bed species tended to decrease due to the natural flooding (Supporting Information Figure S1).

3.3 | Effects of prerestoration conditions and natural floods on species composition

In the NMDS (Figure 5a), the four sections were separated in all years and PERMANOVA revealed significant differences in species composition between sections ($F = 5.7$, $p < 0.01$), but not between years ($F = 0.9$, $p = 0.69$). This was especially valid for Section 2 and two transects of Section 4. Species composition in Section 3 and 5, in contrast, was more similar to each other. Still, except for transects 3C and 4B, all transects were clearly separated from other transects, also during the development from 2011 to 2014 (no crossing of trajectories). The first axis showed positive correlations to total species richness, the proportion of target species, the percentage of indicators of fluctuating water zone, vegetation height and the number of plots. It correlated negatively with the percentage of hardwood species and the proportion of nontarget species. The second axis showed positive correlations with litter cover and the percentage of reed-bed species. NMDS did not reveal similar successional directions of the single transects between the years. However, for most transects (2A, 2C, 3A, 4B, 4C, 5A, 5B and, partly, 2B and 4A), the development between 2012 and 2013 (after the natural flood) and in the following year 2014 without any flooding was just the opposite direction in the NMDS (Figure 5b). For 3B, 3C and 5C, the effects of the natural flood on species composition also continued in 2014 (same direction in NMDS).

4 | DISCUSSION

4.1 | Effect of a natural flood on plant target species

In our study, riparian plant species composition along the streambank in the Danube floodplain showed a high diversity already 1 year after the reconnection to the Danube. Göthe et al. (2016) found that river widening was the most effective river restoration measure, while small-scale measures were less effective. In contrast to this and many other studies (Bauer et al., 2018; Januschke et al., 2014; Jähnig et al., 2009; Modrak et al., 2017), no measures were carried out in our project area that could have increased the extent of gravel or sand bars. Nevertheless, the restoration objective to increase the number of target species was achieved. This is not surprising, as numerous target species were already present in the study area, especially in Sections 3 and 5

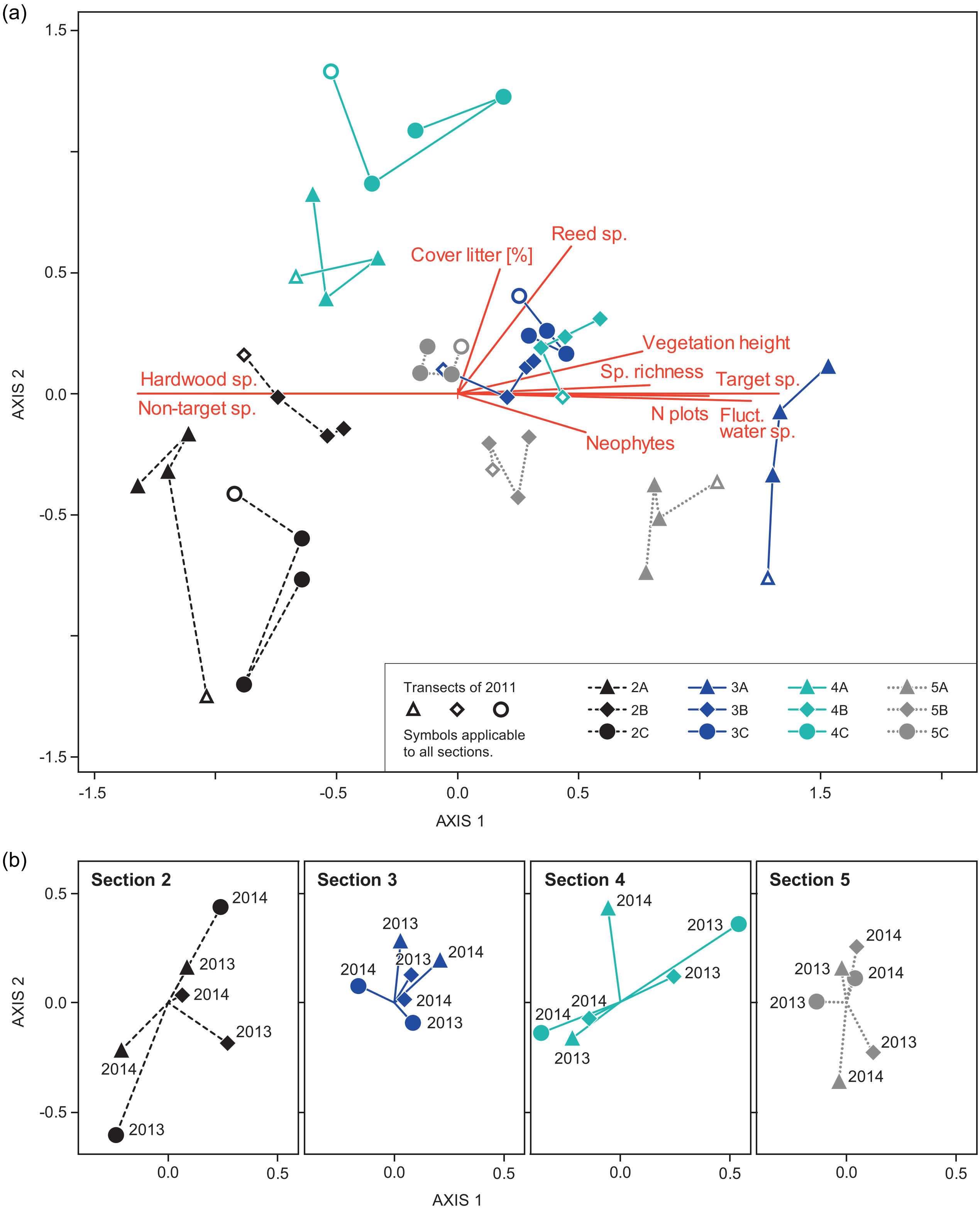

FIGURE 5 Nonmetric multidimensional scaling (NMDS) of the 12 transects in the different sections for 2011, 2012, 2013 and 2014 (connected by trajectories) based on species composition. Stress 12.2%, 90 iterations, r^2: 0.516 (1st axis), 0.235 (2nd axis). (a) Original NMDS with correlation ($r > 0.4$) of structural parameters or species number. Open symbols show transects of 2011 to demonstrate the temporal development; sp.: species (as percentage of all target species, for nontarget and target sp. percentage of total species number), Fluct.: fluctuating; N: number of; (b) NMDS trajectories between the years 2012–2013 and 2013–2014 for the four sections, centred to the origin, indicating the direction of the development of species composition between the years. Changes between 2011 and 2012 are not drawn. Legend is valid for both (a) and (b)

with temporary waters and in the seed bank (Schwab & Kiehl, 2017). Additionally, seed transport within the watercourse promoted the exchange of species between the different sections (Schwab et al., 2018). The only habitat type that was not promoted by the restoration in our study area was the softwood riparian forest. Its target species require very specific site conditions to establish, which were not met during our investigation period: bare unshaded ground with sufficient soil moisture, seed transport by flood after seed shed followed by no further disturbance in the same year (Mosner et al., 2011). Probably more time or even higher flood disturbances than the observed flood in 2013 would be needed to create larger sand and gravel banks close to the waterline or severely suppress reed-bed cover for softwood forest species recruitment.

The other target species groups were able to grow along the streambank and increased after several ecological floodings in 2012 in all sections. The effect of the natural early summer flood in 2013 was less clear. The cover of reed-bed species and the total herb layer were reduced by the higher natural flood, resulting in an increase of fluctuating water zone species in at least a few sections in late summer 2013 after the flood. Nontarget species without any adaptation to flooding stress (Garssen et al., 2015; Merritt et al., 2010; Sterk et al., 2016) are probably showing the most promising response to restoration. The timing of the flooding in early summer might have stronger effects on plant species growth, the survival of existing species and dispersal and germination of newly occurring species than a winter or spring flood (Greet et al., 2011). Particularly, dominant reed species, which have already completely sprouted at that time, can be significantly thinned out by such a summer flood. Although reed beds represent later succession stages (Bauer et al., 2018), we see no need to assess them as less valuable than habitat types of earlier stages, as long as earlier successional stages like fluctuating water zones are still present.

In contrast to the results of Renöfält et al. (2005), the disturbance due to the natural flood had a weak effect on species richness in our study, but high species-turnover rates (up to one-half of all species) were observed. Together with the nonlinear development between years according to NMDS, this reflects that the ecosystem is resilient to the fluvial disturbance (Sterk et al., 2016) and that the target species are adapted to flooding (e.g., resprouting and seed characteristics to quickly respond to occupy suitable conditions after disturbance; Garssen et al., 2015). The natural flood did not accelerate the already successful restoration of streambank restoration in our case, but obviously, progressive succession was interrupted as indicated by an opposite development according to NMDS. Flooding suppressed the dominance of competitive reed-bed species and offered niches for earlier successional stages (i.e., fluctuating water zone). Thus, floods with higher magnitudes, at least like the 10-yearly event here, can prevent the succession of restored sites and are important to preserve the full range of targeted habitat types.

4.2 | Effects of preconditions of the sections on species composition

In our study, the effects of inclination and hydrological prerestoration conditions dominated over the effects of the hydrological events. Modrak et al. (2017) and Göthe et al. (2016) emphasised the importance of specific environmental settings in restoration planning but did not consider differences and heterogeneity at the local scale. The formerly dry Section 2 differed from the other sections in terms of all structural and abiotic parameters as well as species richness and composition, which was also found for the seed bank (Schwab & Kiehl, 2017). Here, a successful establishment of target species will probably need additional time, which only long-term monitoring could confirm. Due to the steep inclination, mainly no floodplain species or hardwood species, which are less adapted to longer flooding periods, were observed along the streambank, but they will presumably be substituted in the future by more fluctuating water zone species on the newly accumulating soils comparable with the effects of river-widening (Göthe et al., 2016).

The other sections were not subject to morphological, but only to hydrological changes. The prerestoration conditions still resulted in different responses of the herb layer to the natural flood. This underlines the findings of Garssen et al. (2015) that responses to flooding are complex and specific. The flat and wide riparian area of Section 3 was comparable with the rivers restored by widening in the studies of Januschke et al. (2014) and Jähnig et al. (2009). Here, herb layer cover even increased due to the natural flood. In the former lentic Section 4 with medium inclination, in contrast, the herb layer cover was reduced by the natural flood, but the balance of target species was positive and the resilient vegetation was able to respond quickly to the changed conditions. In summary, our results indicate that all investigated parameters, initial conditions, inclination and the associated magnitude as well as the duration of fluvial disturbance, have a significant impact on the turnover rate and on the ability of species to cope with the floods, confirming findings of Renöfält et al. (2005) and Wollny et al. (2019).

4.3 | Lessons learned for monitoring

For adaptative management of floodplain restoration (e.g., discharge amount), projects should be accompanied by well-designed monitoring (Stammel et al., 2012). However, especially in spatially variable and naturally ever-changing floodplains, it is challenging to find the right scale (Staentzel et al., 2018), species group (Dziock et al., 2006; Jähnig et al., 2009) and design for permanent plots (Lang et al., 2013). Multiscale assessment is an important basis for understanding complex changes caused by flooding. Among the different methods, habitat mapping is beneficial to document large-scale changes (Jähnig et al., 2009;

Lu et al., 2019; Staentzel et al., 2018). However, our results confirm that major changes occur at the species level (species turnover), also for the less mobile species group ‘plants’. For an impartial evaluation, target species groups must be defined for which the balance between the years can clearly indicate restoration success or failure. These target species groups should represent different stages of succession or adaptation to flood disturbance. Grouping based on plant communities, as in our case, covers this aspect very well by combining parameters such as plant height, moisture prevalence, strategy and resprout capacity (Bauer et al., 2018).

For the riparian zone directly adjacent to the stream, the spatial dimension considering geomorphological changes is of great importance for monitoring. By measuring the cross‐profiles, we were able to measure the fluvially induced morphodynamics and to relate them to vegetation changes, although a few plots were flushed away by floods and new ones developed by sediment accumulation. Even though transects are difficult to evaluate statistically due to their spatial dependency (Lang et al., 2013), they provided completely new insights into the development of the streambank that can hardly be captured with scattered permanent plots or stratified plot sampling. We were able to determine which species had been lost or newly established even at sites with high levels of morphodynamic change. In contrast, analyses of the individual 1 m^2 plots (standard size for streambanks; e.g., Bauer et al., 2018; Göthe et al., 2016; Renöfält et al., 2005) hardly showed any differences neither between years nor between sections. Analyses of all species of one transect, however, revealed clearer differences. The objective of monitoring is to show the temporal development of species diversity and the success or failure of restoration measures. Thus, analysing the whole width of the riparian zone is more promising than monitoring individual small plots only. Still, interpreting the results must consider that species richness of longer transects is presumably higher than that of short ones. To guarantee statistically sound analyses and justified adaptive management, a sufficient number of transects is required. This is especially valid for those monitoring setups where no control plots are available. The idealised and statistically sound BACI design (Before–After–Control–Impact; Smokorowski & Randall, 2017) can hardly be completed in anthropogenically superimposed landscapes or along newly created streams. In our study, we could only compare the ‘before’ and ‘after’ situation, but not the ‘control’, thus, we added a comparison between the different sections. The significant differences between the sections impressively showed that a space‐for‐time approach is not efficient for controlling the success of a specific floodplain stream restoration that contrasts with studies comparing several rivers (Januschke et al., 2014; Modrak et al., 2017).

This leads to the fourth dimension of particular relevance in floodplains: time (Ward, 1989). Several authors (Bauer et al., 2018; Januschke et al., 2014; Lu et al., 2019) pointed out the importance of repeated monitoring over a longer period. As our study focused on the effects of a decennial flood event that occurred after implementing the restoration measures, we investigated only a period of 4 years. During this time, we observed an ongoing change of species composition, which did not enable us to predict the situation in the next years or decades. Nevertheless, the strong effects of the natural flood underline that event‐related monitoring is even more essential. We were able to show that flooding has a resetting function for species composition to earlier successional stages (favouring indicators of fluctuating water zone and repressing cover and species number of reed beds). Event‐related monitoring should analyse not only floods but also extreme low‐water phases and other climatic events (Stammel et al., 2016). Such monitoring will enable us to understand floodplain processes better and should supplement regular long‐term monitoring.

In conclusion, floodplain restoration is promising and should be monitored in the long term, but especially on an event‐related basis with a set of several species groups (Dziock et al., 2006; Jähnig et al., 2009; Stammel et al., 2012) and scales (Staentzel et al., 2018). Each individual species group offers different insights into the effects of morphological and hydrological dynamics (Januschke et al., 2014) and helps to establish adaptive management. Our study has shown that especially the vegetation of riparian habitats along the streambank with their immense ecological importance is highly resilient to extreme flood events and should always be included in monitoring floodplain and river restoration measures (Modrak et al., 2017). To observe the changes along morphodynamically active streams, cross‐profile transects are the best way to capture the development, but spatial differences of the floodplain (pre‐ and postrestoration) must be reflected when selecting the transects. As streambanks are water‐dependent habitats, they should be integrated into long‐term river monitoring (e.g., within the framework of the Water Framework Directive). Then, additional event‐related monitoring can easily be carried out.

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